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Rivers As Sentinels: Using the Biology of Rivers to Guide Landscape Management

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SUMMARY

- Humans alter the surface of Earth in ways, on scales, and at frequencies unprecedented in recent history. Resource and environmental managers must identify and minimize the effects of changes with negative consequence for human society.

- Because rivers integrate all that happens in their landscapes, their condition, especially their biological condition, tells us much about the consequences of our actions.

- The condition of rivers in the Pacific Northwest tells us that much of the region's rich natural capital has been spent.

- Existing laws do not adequately protect rivers because they are at odds with the physical connectedness of water, and, worse, they commonly ignore the biological components of aquatic ecosystems.

- Human actions jeopardize the biological integrity of water resources by altering physical habitat, modifying seasonal flow of water, changing the food base of the system, changing interactions within the stream biota, and polluting water with chemical contaminants.

- Conventional monitoring and evaluation studies--tracking chemical pollution or population sizes of target species--are inadequate to protect overall river condition in part because they are conceptually narrow, in part because they are not well suited to distinquishing variation caused by natural events from variation caused by human actions.

- Biological monitoring in the twentieth century began with a restricted focus

(organic pollution, toxic chemicals) but is shifting to a more integrative approach that evaluates the condition of aquatic biota from diverse perspectives.

- Integrative multimetric biological indexes should be used to develop biological standards (criteria) because (1) they are more comprehensive and rodust than chemical standards and (2) they are effective at diagnosing degradation, defining its cause(s), and suggesting treatments to halt or reverse damage.

- Multimetric biological monitoring is a central feature of water resource assessments throughout the United States--48 states have (42) or are developing (6) multimetric approaches--and it has been used on all continents but Antarctica.

- The index of biological integrity (IBI) is one multimetric approach that is being applied to examine the influence of humans on fish, invertebrate, and algal assemblages.

- IBI has substantial statistical power to detect the effects of diverse human actions (agriculture, livestock grazing, logging, recreation, and urbanization) and their consequences (point and nonpoint pollution, physical habitat alteration, flow alteration, and complex cumulative impacts) on water resources.

- IBI can be used to define spatial and temporal patterns in water resource condition and to evaluate the effects of management efforts.

- A benthic invertebrate index of biological integrity (B-IBI) proposed for use in the Pacific Northwest includes ten metrics: total number of taxa; number of mayfly, stonefly, caddisfly, long-lived, intolerant, and clinger taxa; proportion of individuals belonging to tolerant taxa and to predatory taxa; and percent dominance of the three most abundant taxa.

- Human actions that deplete and degrade Earth's ability to sustain life--which creates the environment suitable for society--place human society at risk.

- Rivers are sentinels: they give us early warning of the risks our activities engender. We can no longer afford to ignore these risks or behave as if they did not exist.

INTRODUCTION

Environmental change is a reality, and it is continuous. Change on Earth is driven by wind and water; geological forces, astronomical events; and the work of microorganisms, plants, and animals. During the past two centuries, human activities have become the principle driver of change on Earth. Human influences are massive, they are incessant, and they are global. For the first time, a biological agent--a single species--rivals geophysical forces in shaping the surface of the Earth.

Human-caused change may be positive, neutral, or negative. The challenge faced by environmental scientists is to distinguish among these three alternatives by detecting and interpreting the causes and consequences of change, especially those that alter living systems. Resource and environmental managers want to detect and treat changes that have negative consequences; at the same time, they want to avoid wasting resources treating changes that do not have negative consequences.

RIVERS AS SENTINELS

Rivers and streams serve as a continent's circulatory system, and the study of those rivers, like the study of blood, can diagnose the health not only of the rivers themselves but of their landscapes (Sioli 1975). Changes anywhere on the landscape are likely to influence rivers. People change the landscape as they harvest forests. They compromise river health when they build dams to generate power or control flooding, or when they mine lands for minerals. They degrade rivers when they construct industrial parks to manufacture goods, shopping centers to sell those goods, homes for families, and farms and tree plantations to supply food and fiber. They degrade rivers when they construct transportation corridors to bring people and goods together.

Before Pacific Northwest rivers and landscapes were altered by these activities, humans in the region had prospered for thousands of years on its wealth of forest, river, and marine resources. When European colonists arrived about 200 years ago, they altered forever the interactions between regional natural systems and their human populations. Now the Northwest's demand for water exceeds supply; its forests and fish populations are decimated; its soils are eroded; and its marine resources are damaged and depleted. These and other changes in the health of Northwest landscapes indicate that modern stewardship of regional resources has fallen short. The region's rich natural capital has been squandered.

We have only to look at Pacific Northwest rivers, and especially their living inhabitants, to see signs of degradation. Plummeting populations of migratory salmonids (Nehlsen et al. 1991) and other, nonanadromous native fishes, for example, follow directly from dams, excessive water withdrawals for irrigation, and other destruction of habitat; they are the consequence of overharvest by sport and commercial fishers and of the effects of invading exotic species. Even when society tried to protect native fish stocks, the fishes often continued to decline. To offset fishery losses from overharvest and habitat destruction, stocking streams with fish raised in hatcheries began in the 1800s (Bowen 1970), but for more than 100 years, it has been obvious that fish stocking exacerbates the decline in wild salmonid populations (White et al. 1995).

Few are so naive to assume that the changes people cause can be stopped altogether. But it is equally naive to suggest that these changes can proceed without consequences. Laws, regulations, and management programs have proliferated to reduce human damage to river systems, but too often, society's interests in water resources have still not been protected. American legal doctrines such as "prior appropriation" in the West and the Clean Water Act are simply at odds with the physical connectedness of water; worse, they completely ignore the biological components of aquatic ecosystems (Karr 1990, 1995a, Johnson and Paschal 1995). Water in rivers is connected in four dimensions: upstream and downstream (longitudinally); across channels, hyporheic, and groundwater zones (vertically); on the surface from uplands through riparian corridors to the channels (horizontally); and through the water cycle from clouds to precipitation to surface and then ground water (temporally).

Existing laws and their implementing regulations treat water narrowly--as if surface and groundwater were not connected, as if point and nonpoint sources of pollution could be treated in isolation. They undervalue the immense diversity of goods and services supplied by aquatic ecosystems. Improvements in the laws--and in their implementing regulations--are urgently needed if we are, as the Clean Water Act mandates, "to restore and maintain the physical, chemical, and biological integrity of the nation's waters."

For 25 years, this mandate was largely ignored in water policy (Karr and Dudley 1981, USEPA 1990, Karr 1991). Three approaches to the use and management of water resources kept the focus narrow, incomplete, and inadequate:

1. *Water was viewed as a fluid for humans to use.* Too many water resource professionals saw "the forms of life in a river [as] purely incidental, compared with the main task of a river, which is to conduct water runoff from an area toward the oceans" (Einstein 1972).

2. *Pollution was the only threat to water resources, and dilution was the solution.* People managed for "water quality" (degrees of chemical contamination). In 1965 an Illinois water official observed, "Regardless of how one may feel about the discharge of waste products into surface waters, it is accepted as a universal practice and . . . a legitimate use of stream waters" (Evans 1965). Surface waters existed to receive the discharge of human society.

Neither of these two attitudes gave any value to the life forms associated with aquatic ecosystems.

3. *Only a few aquatic species "counted" as important to human society.* Society sought to maximize sport or commercial harvest of selected species. Production--larger harvests of fish or shellfish--became the goal, and technofixes like hatcheries became the means to supplement falling wild populations (Meffe 1992). Fish ladders helped migrating adults pass upstream over dams, but no provisions were made for helping young fish go around the dams as they migrated downstream toward the ocean. Many biologists removed large woody debris from stream channels to make passage easier--never mind that fish had been passing such barriers for centuries, or that the wood actually created fish habitat (Maser and Sedell 1994).

Although these three philosophies have not been abandoned, a growing number of water resource professionals recognize their inadequacies. Citizen watershed teams and government agencies now recognize the importance of rivers and are struggling to establish long-term monitoring and assessment programs. Weber (1981) argued that the only way to determine the biological integrity of aquatic assemblages is to examine their properties in the field and to compare them with assemblages expected in the absence of degradation caused by human actions. More and more, water managers are being called upon to evaluate the biological effects of their management decisions, for no other aspect of a river gives a more integrative perspective about the condition of a river than its biota.

Biologists have intuitively applied this fact for decades, but usually by focusing on "pollution tolerance" or on population sizes of a few target species. The first approach concentrated on the ability of individuals to survive stress from pollution, especially organic contamination (Kolkwitz and Marsson 1908). Even water resource biologists rarely looked at phenomena other than chemical contamination. Fishery managers, on the other hand, tracked the population sizes of species with recreational or commercial importance.

Agencies charged with reducing pollution (departments of environmental quality) and population enhancement (departments of fish and game) developed independently in agencies that rarely interacted, even though they "managed" the same water body. Each organization's programs were carried out as if scientists knew how organisms would respond to their management prescriptions; monitoring to see whether a program was actually effective occurred rarely or not at all. Failure to evaluate management decisions allowed wasteful, even destructive, practices to continue for decades (Maser and Sedell 1994, White et al. 1995).

In short, water resource managers either ignored biological systems or implemented policy with only narrow conceptions of biological conditions and their importance to human society. Reductionist viewpoints dominated water management; legal and regulatory programs avoided most biological issues and contexts; precise biological goals were not well developed or defined; field methods to measure biological condition were not standardized; formal processes to evaluate and express biological condition were not established; links between field measurements and enforceable goals were weak; and approaches to measuring biological condition were not cost effective (Karr 1991).

By now, more-comprehensive approaches have been developed and are being adopted by state and federal agencies. Forty-two states now use multimetric biological assessments of biological condition and six states are developing biological assessment approaches; only three states used multimetric biological approaches in 1989 (USEPA 1996a). Efforts are at last being made to monitor the biological integrity of water resources as mandated by the Clean Water Act 25 years ago (Karr 1991, Davis and Simon 1995, USEPA 1996a,b).

CUMULATIVE EFFECTS AND BIOLOGICAL INTEGRITY

Human impacts on the biological integrity of water resources are complex and cumulative. Human actions jeopardize the biological integrity of water resources by altering one or more of five principal sets of factors: alteration of physical habitat, modifications of seasonal flow of water, changes in the food base of the system, changes in interactions within the stream biota, and chemical contamination (Karr et al. 1986, Karr 1991). When humans modify landscapes or stream channels, they alter one or more of these attributes, and changes in biological integrity are likely (Table 1). These features are connected--physically, chemically, and biologically. Furthermore, the biota of streams evolved in the presence of, and now depends on, variability in time and connectivity throughout landscapes (Meyer et al. 1988, Pringle et al. 1988, Naiman 1992). Attempts to protect water resources must take account of all these factors and the cumulative effects that human actions impose on them.

The biota that evolves and maintains itself in a region possesses biological integrity (Karr and Dudley 1981, Angermeier and Karr 1994, Karr 1996). Biologial integrity refers to the capacity to support and maintain a balanced, integrated, and adaptive biological sysem having the full range of elements and processes expected in a region's natural habitat. Critical elements comprise genes, populations, species, and assemblages; critical processes encompass mutation, demography, biotic interactions, biogeochemical cycles, energy dynamics, and metapopulation processes. Together they create the regional complex of living organisms and act over a variety of spatial and temporal scales. Living systems thus are embedded in dynamic evolutionary and biogeographic contexts (Karr 1996).

Systems possessing biological integrity can withstand, or recover rapidly from, most *natural* disturbance (Rapport et al. 1985, Karr et al. 1986). Biological integrity declines if the natural disturbance regime is altered by a type, intensity, or frequency of perturbation that lies outside the biota's adaptive experience, especially if those disturbances become incessant. Urbanization, for example, compromises the biological integrity of streams by severing the connections among segments of a watershed and by altering hydrology, water quality, energy sources, habitat structure, and biotic interactions.

The only practical and cost-effective approach to determine if human actions are degrading biological integrity is systematic biological monitoring and assessment (Davis and Simon 1995). Such monitoring provides both numeric and narrative descriptions of resource condition, which can be compared among watersheds, across a single watershed, and over time (Karr 1991), and it does so at costs that are often less than the cost of complex chemical monitoring.[1]

EVOLUTION OF BIOLOGICAL MONITORING

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[1] Costs per evaluation are low for ambient biological monitoring (based on a decade of sampling and including equipment; supplies; and logistical, administrative, and data analysis and interpretation activities: benthic invertebrates, \$824; fish, \$740; Yoder and Rankin 1995) in comparison with chemical and physical water quality (\$1,653) and bioassays (\$3,573 to \$18,318).

Ecological systems, especially their biological components, are exceedingly complicated, and they have been studied from diverse perspectives. Theoretical ecologists try to understand natural variation within assemblages of organisms over space and time, along with the evolutionary and thermodynamic principles that mediate this variation. They ask questions such as, Why does the number of species vary from place to place on the surface of the Earth? What regulates the size of animal and plant populations? How do global biogeochemical cycles regulate ecosystem structure and function? For the most part, they work in natural systems subjected to relatively little influence from human actions.

Applied ecologists need to recognize natural variation and, in addition, to understand how natural systems respond to stresses imposed by human society. They ask, What do we measure to understand responses to stress? How do we interpret the results? How do we distinguish natural variation from human-induced change? What are the likely consequences of the changes we see? How do we tell citizens, policymakers, and political leaders what is happening and how to fix it?

Biological monitoring has evolved rapidly during the twentieth century as knowledge of aquatic ecosystems has changed and human-imposed stresses have become more complex and pervasive. Early water quality specialists developed *biotic indexes* sensitive to organic pollution (sewage) and sedimentation (Kolkwitz and Marsson 1908); this approach continues in modern "biotic indexes" (Chutter 1972, Hilsenhoff 1982, Lenat 1988, 1993). Then as toxic chemicals became more pervasive, water managers recognized the limitations of early biotic indexes and began to screen for the biological effects of synthetic as well as "natural" chemical contaminants. Biologists exposed fish or invertebrates experimentally to contaminants and documented toxicological dose-response curves. For a given body size, they observed, very low doses of a chemical contaminant might lead to little or no response. As dose increased, response also increased. The goal was to establish quantitative *chemical criteria* to use in water quality standards, criteria that would protect human health or populations of desirable aquatic species.

But just as biotic indexes measure largely the effects of organic pollution, chemical criteria based on toxicology generally address only a small number of chemical contaminants. Chemical criteria based on dose-response curves for single toxicants do not incorporate synergistic or other interactions of multiple chemicals in the environment. Further, the toxicological approach ignores other human impacts on aquatic biota, such as habitat alteration.

At the same time, biologists responsible for sport and commercial fisheries built hatcheries (White et al. 1995); "enhanced" habitat by first removing, and later adding, coarse woody debris (Maser and Sedell 1994); and introduced exotic species as alternatives to native fisheries (Courtenay and Moyle 1992, Allan and Flecker 1993). These "population enhancement" efforts are now recognized not only as inadequate but, in many circumstances, as damaging to regional biological resources.

In a recent review, Fausch et al. (1990) identified four major measurement approaches commonly used to detect and understand the effects of human actions on aquatic organisms: indicator taxa or guilds; species richness, diversity, and evenness; multivariate statistics; and multimetric indexes such as the index of biotic integrity (IBI: see reviews in Karr et al. 1986, Davis and Simon 1995, Fore et al. 1996).

The most common approach is to track human-induced *change in abundances* (or population size or density) of indicator taxa or guilds. Managers and scientists conducting field assessments of environmental impacts must isolate the effect of interest from "noise" caused by natural spatial and temporal variation (Osenberg et al. 1994). Biological studies have not accomplished this task well in the past when compared with physical or chemical parameters. Data from long-term studies of marine invertebrates (Osenberg et al. 1994), for example, show that temporal variability in population-based parameters (e.g., densities of organisms) show about three times the variability of individual-based parameters (e.g., size and body condition of individuals) and nearly four times the variability of chemical-physical parameters (e.g., water temperature, sediment quality, water column elements). In my own experience, repeated sampling of standard water quality parameters in streams and rivers typically yields coefficients of variation (CV) on the order of 20-25% while CVs for standard measures of population size are much larger at 50-300% in standard sampling designs.

Early efforts in field assessment used "control-impact" (CI) or "before-after" (BA) sampling designs, which often do not detect or reveal patterns relevant for water resource management. To overcome this problem, Green (1979) proposed a single design combining the two ("BACI") to separate the effect of human activity from other sources of variability in space or time. Because the BACI design confounds interactions between time and location, however, still other statistical approaches were proposed: "before-after-control-impact paired series" (BACIPS; Stewart-Oaten et al. 1986) and "beyond BACI" (Underwood 1991, 1994). Understanding the interaction of variability and effect size is also critical to determine statistical power in planning and interpreting environmental impacts (Osenberg et al. 1994). The growing sophistication of these designs demonstrates the effort being made to improve field assessment protocols. But their complexity also goes beyond the planning, sample size, and analytical capability of most routine monitoring efforts; their narrow biological contxt limits managers ability to detect other relevant biological signal. (See Schmitt and Osenberg [1996] for an excellent review of these sampling designs and their use.)

During the 1960s, ecological research embraced species diversity as a central theme, so measures of diversity were used to evaluate biological communities. More than 25 years ago, Hurlbert (1971) raised concerns about the statistical properties of *diversity indexes*; others have questioned their biological properties (Wolda 1981, Fausch et al. 1990). Diversity indexes often respond erratically to systematic changes in assemblages; they are often inconsistent and dependent on initial conditions--all of which leads to ambiguous interpretations (Wolda 1981, Boyle et al. 1990). These measures were nevertheless advocated for use in water management (Wilhm and Dorris 1968) and used in a few cases (e.g., to set Florida water quality standards, although Florida is now moving away from use of diversity indexes to multimetric evaluations; Barbour et al. 1996). Few scientists and managers recommend species diversity indexes today, largely because biologically more reasonable and statistically more reliable approaches are available. Unfortunately, however, these diversity indexes left a negative semantic legacy that surfaces whenever the word *index* appears (Suter 1993 is an example of that legacy).

Many researchers advocate *multivariate statistical approaches* because they ostensibly provide an objective way to explore variation in biological data. Principal components analysis (PCA), the most common procedure (James and McCulloch 1990), and other ranking techniques attempt to extract maximum statistical variance from variance-covariance matrices, usually across species or sites (Ludwig and Reynolds 1988). The assumption is that describing the maximum variation will identify the most meaningful signal about biological condition. Ordination, another multivariate procedure, was developed for data that follow a multivariate normal distribution (Fore et al. 1996), a rare pattern in data from biological monitoring.

Although multivariate methods may seem to provide an objective view of data, they often lose valuable information. Many users of multivariate methods (1) focus on presence and absence of taxa or numbers of animals present, rather than knowledge of natural history or organismal responses to human actions; (2) emphasize statistical (variance-covariance) rather than biologically relevant patterns and their consequences; and (3) exclude rare species from analyses in an effort to avoid the effect of zeros in a data matrix. For example, one participant in the Ninth Annual Technical Information Workshop on study design and data analysis in benthic macroinvertebrate assessments at the North American Benthological Society meeting (June 1996) concluded that rare species simply add "noise to the community structure signal and . . . little information to the data analysis, and therefore, rare species should be excluded from the data matrix. We recommend excluding all taxa that contribute less than 1% of the total number or occur at less that 10% of the sites" (Reynoldson and Rosenberg 1996, p. 5, also see Norris 1995). Such exclusion represents a substantial loss of important biological information.

Without objective statistical tests to determine what is different--and the biological consequences of those differences--multivariate analyses rarely give results that go beyond commonsense knowledge (Karr and Martin 1981, Stewart-Oaten 1996, Fore et al. 1996). Gotelli and Graves (1996, p. 137) go as far as to suggest that "multivariate analysis has been greatly abused by ecologists." They further note that the practice of "drawing polygons (or amoebas) around groups of [points], and interpreting the results often amounts to ecological palmistry. Ad hoc 'explanations' often are based on the original untransformed variables, so that the multivariate transformation offers no more insight than the original variables did."

Gotelli and Graves conclude that patterns in multivariate analysis should be compared against a properly formulated null model. Comparing the results of PCA analysis of real data with similar matrices of random numbers, however, shows that (1) the percent of variation described may be similar for both, especially for the second and higher principal components; (2) loadings of original variables on principal axes are often as high for random numbers as for real data; and (3) matrix size is an important determinant of the amount of variation extracted (Karr and Martin 1981). Multivariate techniques were unable to discern known deterministic relationships in one study (Armstrong 1967), and they manufactured relationships in data sets containing no such relationships (Rexstad et al. 1988).

Used cautiously, multivariate statistics can point to patterns when little is known about the underlying natural history of a biota (Gerritsen 1995). But because we know much about streams and landscapes, invertebrates and fish, and the effects of humans on those places and organisms, I prefer an approach that actively and explicitly uses that knowledge.

Multimetric indexes use biological knowledge; they also use knowledge from earlier monitoring approaches, integrating that information into a single method while avoiding theoretically flawed indicators (e.g., species diversity indexes). They are also wider in scope (Davis 1995, Simon and Lyons 1995). The set of metrics (or biological attributes) incorporated into a multimetric index integrates information from ecosystem, community, population, and individual levels (Karr 1991, Barbour et al. 1995, Gerritsen 1995). Multimetric indexes are generally dominated by metrics of taxa richness (number of taxa) because structural changes, such as shifts among taxa, generally occur at lower levels of stress than do changes in ecosystem processes (Karr et al. 1986, Schindler 1987, 1990, Howarth 1991, Karr 1991). But the most appropriate and integrative multimetric indexes explicitly embrace several concepts, including taxa richness; indicator taxa or guilds (e.g., tolerant and intolerant); health of individual organisms; and assessment of processes (e.g., as reflected by trophic structure or reproductive biology) of the sampled assemblage).

Like the multimetric indexes used to track national economies (Mitchell and

Burns 1938), multimetric biological indexes measure many dimensions of complex ecological systems. Multimetric economic indexes assess economic health against a standard fiscal period; indexes of biological integrity assess the biological well-being of sites against a regional "baseline condition" reflecting the relative absence of human influence.

Stream ecologists recognize that stream segments (or habitat patches)--even those in the same region--may not share the same evolutionary and biogeographic history, and thus the expectations of biological condition are not the same at all places. Thus classifying system types (e.g., stream size or stream type) for a given region is essential in defining baseline conditions within the region; the conditions reflecting the absence of human influence in one patch may be quite different from baseline conditions in another. Again, the goal is to understand and isolate, through sampling design and analytical procedures, patterns that derive from natural variation in environments. Too many existing studies confound these patterns with human-induced variation, making interpretation of biological signal difficult or impossible.

THE INDEX OF BIOLOGICAL INTEGRITY

The index of biological integrity (Karr 1981) was the first comprehensive multimetric index applied to assess biological condition in running waters. When used properly, IBI (1) detects degradation of living systems; (2) diagnoses the likely causes of degradation; (3) identifies management actions that can halt or reverse degradation; and (4) monitors living systems to find out if management efforts to restore degraded sites are succeeding. The foundation for IBI was established in a project in Allen County, Indiana, that began in 1973, soon after passage of the 1972 Clean Water Act (PL 92-500). Most project participants concentrated on chemical benchmarks; some, including US Environmental Protection Agency officials, even denied the relevance of a biological perspective.

Finding the chemical approach inadequate, I attempted to track resource condition based on the resident fish assemblage. The resultant index incorporated metrics reflecting species richness, indicator taxa, trophic guilds, fish abundance, presence of exotic species, and condition of individual fish (Table 2). Efforts to adapt this fish IBI to benthic invertebrates have proposed similar metrics.[2]

[2] The extent to which proposed metrics were evaluated before protocols were published varied from high for the invertebrate community index (ICI; Ohio EPA 1988, DeShon 1995) and the benthic index of biological integrity (B-IBI; Kerans and Karr 1994, Fore et al. 1996) to low for rapid bioassessment protocol III (RBP-III; Plafkin et al. 1989).

To successfully apply multimetric biological indexes, one must: (1) select measurable attributes that provide reliable and relevant signal about the effects of human activities, (2) develop sampling protocols and designs that ensure that those attributes are measured accurately; (3) define analytical procedures to extract and understand relevant pattern in the sample data; and (4) communicate those results to policymakers and to society so that all stakeholder communities can contribute to the development of environmental policy. Not all multimetric indexes have been successful. Failure generally stems from inadequate attention to one or another of these four activities.

An infinite variety of biological attributes could be measured; a much smaller number of attributes provide useful signal about the impact of human activities on local and regional biological systems. Some attributes vary little or not at all (e.g., the number of scales on the lateral line of a fish); others vary substantially (e.g., weight, which varies with age or environmental context). Variation may be natural or human-caused. Natural variation derives from temporal (diurnal, seasonal, annual) and spatial (stream size, stream channel type) sources. Thus the first step in IBI development is to define how humans influence aquatic ecosystems (Karr et al. 1986, Karr 1991; see Table 1) and what biological attributes can be measured with precision to provide reliable information about biological condition.

Selecting IBI metrics

Before toxicologists could establish chemical criteria, they documented the response of organisms to the presence of specific contaminants in experimental conditions and established dose-response curves for those contaminants acting on the same organisms (Fig. 1, Curves A and B). Similarly, before selecting a metric for inclusion in a multimetric index such as IBI, one must find an empirical relationship between the attribute proposed for measurement and a gradient of human influence. Unlike the contaminant-specific dose-response curves of toxicology, the ecological dose-response curves used to define metrics for an IBI reflect specific human activities that exist in a set of study watersheds. Curves C through F (Fig. 1) are cumulative ecological dose-responses for several biological attributes along a gradient of human influence (e.g., urbanization in a study watershed): predator taxa richness (C), total taxa richness (D), intolerant taxa richness (E), and % of individuals in tolerant taxa (F).

Study watersheds can be carefully selected to involve a gradient of only one human activity (e.g., area logged or grazing intensity), or they may reflect the cumulative impacts of many human activities (e.g., chemical contaminants, varied land uses, riparian condition). These ecological dose-response curves are, in essence, empirically derived descriptors of the responses of biological systems to human-induced stress. Because of natural variation in the range of human impact associated with a given value of the biological attribute on the y-axis, a single metric only generally defines the level of human impact (Fig. 1, ellipses G and H). Simultaneous use of multiple metrics narrows the identified degree of biotic integrity (or amount of degradation) on the x-axis.

Regardless of whether fish, invertebrates, or other taxa are used, the search for a small set of biological attributes that reliably signal resource condition along gradients of human influence yields the same basic list of attributes (Miller et al. 1988, Karr 1991, Barbour et al. 1995, Davis and Simon 1995). With only minor tuning, the list can be adapted to specific regions (Miller et al. 1988). Taxa richness of darters in midwestern North America is replaced by taxa richness of sculpins in the Pacific Northwest, for example; both metrics reflect the presence or absence of riffle-dwelling benthic insectivores. In the only application of the fish IBI to the Pacific Northwest, Hughes and Gammon (1987) used half the midwestern IBI metrics (total number of native fish, sucker, and intolerant species; proportion omnivores; and abundance) and modified others (number of darters replaced by sculpins, sunfish by minnow species; proportion of green sunfish replaced by carp, top carnivores by catchable salmonids (>20 cm), and hybrids by introduced species).

Six recent studies of stream invertebrates in three regions (the Tennessee River valley, Pacific Northwest, west-central Japan) suggest 10 metrics, out of about 30 routinely tested, as reliable indicators of human impact in diverse circumstances (Table 3). Further, these metrics seem robust regardless of sampling method (e.g., Surber samplers, Hess samplers, or kicknets).

Metric selection is based on informal, especially graphical, methods rather than formal statistical hypothesis testing (Fausch et al. 1984, Fore et al. 1996). Formal methods can "obscure the information in the data rather than clarify it" because they are interpreted as what should be done rather than as judgments about what can be done (Stewart-Oaten 1995).

In the Northwest, best results are obtained from sampling protocols with three replicate samples taken from a single riffle during September (Fore et al. 1996). Choice of a standard sampling time avoids the need to interpret variation because of seasonal patterns in arthropod abundances and distributions. Surber samples provide an excellent foundation for biological assessment, but kicknet samples may also be used. The important point is that samples must be collected and processed with carefully defined standard procedures; samples from different sites or times should not be mixed. Results improve if each sample replicate contains at least 400 fish (Fore et al.

1994) or 500 invertebrates, all of which are counted.

Scoring metrics

The different biological contexts for different metrics produce different quantitative ranges for each; proportions of samples range from 0 to 100%, and taxa richness may vary from 0 to a few taxa (stoneflies) or 0 to 40 (total taxa richness). Thus, before one can combine metric results into a single integrative index, one must convert the data into a common scoring base. Whether invertebrates or fish are sampled, each site must be compared with a reference standard for each metric. By the convention established in the original IBI (Karr 1981), a site receives a score of 5 if a metric value lies at or near the value expected at a site minimally altered by humans, 3 if moderately degraded, and 1 if severely degraded.

Defining scoring criteria requires some understanding of normal biological responses to human activity. In the case of species (or taxa) richness, one expects a monotonic decline (or increase) in number of species present as human influence increases. Taxa richnesses are generally trisected (Fig. 2a,b): if 27 species are expected at an undisturbed site, for example, the score is 5 if 19 to 27 species are present, 3 with 10 to 18 species, and 1 if 0 to 9 species in a standard sample. More complex relationships are often scored as appropriate to the structure of the relevant "dose-response" curve (Fig. 2c,d: Karr 1992, Kerans and Karr 1994, Fore et al.

1996). Because species richness for most taxa varies with stream size, lines defining maximum expected species richness across a gradient of stream size is appropriate in each sampling region (Fig. 3, Fausch et al. 1984).

Integrating multiple metrics

Comparisons among sites in space and time are easier if both narrative and numeric expressions of resource condition exist. The challenge is to combine the substantial information contained in each individual metric into an easily interpreted and communicated quantitative expression about the relative condition of sites within a region or about the same site over time. That quantitative expression is IBI, the sum of the converted metric scores (5, 3, or 1). Detailed information about individual sites, including the status of individual metrics, is not lost because the components of the multimetric index can still be examined individually in both quantitative and narrative form (Simon and Lyons 1995). With regard to index variability, both empirical and statistical evaluations of variation in IBIs across time at numerous sites suggests that variation at sites with little or no change in human influence over time is on the order of 8-10% (Fore et al. 1994).

Managers can use the overall IBI score, or a narrative description of each biological attribute of a site, to communicate with policymakers or citizens. The existence of numeric and narrative descriptions makes it possible to compare sites, thereby permitting more effective analysis of biotic condition. Policymakers can then establish priorities for protection or clean up. Because metric responses along gradients are essentially dose-response curves, managers can also predict the effects of certain proposed actions, when evaluating permit requests, for example. Knowing general patterns of biological response makes it possible to predict the consequences of alternative actions, thereby leading to better management decisions. That knowledge also makes it possible to predict degradation that might be expected following increased human activities in a watershed or, alternatively, the improvement expected when certain activities are curtailed.

When used correctly, the IBI approach costs no more, and often less, than conventional chemical screening; it is sensitive to the full array of human impacts on living systems (see "Cumulative Effects and Biological Integrity" above); and it can provide sophisticated and interpretable data in a short time.

WHAT IBI SAYS ABOUT STREAMS AND WATERSHEDS

IBIs have now been defined for a number of taxa (fish, insects, algae, birds); habitat types (streams, wetlands, estuaries); and geographic regions (all continents but Antarctica). As the following examples illustrate, IBIs can inform a variety of assessment and management contexts.

Detecting the effects of point-source pollution

IBI successfully detected variation in biological condition (aquatic invertebrates) along the length of a stream in the North Fork Holston River in Virginia and Tennessee (Fig. 4). Four years of data on invertebrates yielded high benthic IBI values upstream of a sludge pond. Immediately below the sludge pond, B-IBI dropped sharply, then increased slowly downstream from the pond. Although B-IBI values varied over the years, rankings of sites along the stream remained strikingly consistent across the four-year sampling period.

Identifying multiple sources of degradation

Most streams are influenced by several human activities. One important test of IBI is its ability to detect different human impacts. Big Ditch is a third-order stream in a largely agricultural watershed (90% row crops) in east-central Illinois (Fig. 5a). The regional landscape is degraded, and so is the stream channel--no more than a homogeneous raceway with sand, gravel, and rock substrates and no riparian vegetation. A municipal sewage treatment plant just above sampling station 2 pours 75 million liters per day of wastewater into the stream. The fish IBI was moderate above, and low below, this effluent source. IBI increased slowly along the stream from stations 2 to 4 (recently channelized) and rose sharply at stations 5 and 6 (not recently channelized). Point-source inputs, time since channelization, and intensive regional land use interacted to produce a complex longitudinal pattern of biological condition along this single stream; IBI detected those differences.

Describing geographic pattern and detecting cause

Sample sites within the Raisin River, Michigan, show different levels of biological integrity among the townships within the watershed (Fig. 5b). Areas with low IBI values (fish) are associated with larger towns, extensive agricultural areas, and feed lots. Geographic analysis informs managers where degradation is worst, diagnoses the causes of degradation, and suggests where regulatory actions or incentive programs are required to improve water resource condition.

Detecting regional variation in human influence

Degrees of degradation from human actions in different regions can be detected with IBI (Fig. 5c). IBI scores (fish) were high for five sites in Arkansas selected as representing the best conditions in each major region of the state. IBI ranked as good or better 92% of sites in the Red River, Kentucky, which runs through the Daniel Boone Wilderness. Sites sampled in the Raisin River, Michigan, and the seven-county area around Chicago, Illinois, ranged from excellent to severely degraded (no fish). Average IBI scores for the heavily urbanized Chicago area were well below those of the Raisin River.

Detecting change over time as human activity changes

A small woodlot in Wertz Drain, Allen County, Indiana, had a high fish IBI, reflecting good habitat quality (sinuous channel, well-developed pools and riffles, trees shading the channel in a good riparian corridor). A badly done stabilization of the bank upstream of the woodlot in 1976 delivered substantial sediment downstream, which filled pools and degraded habitat within the woodlot. The resident fish community deteriorated, then recovered slowly over the next five years (Fig. 6a).

Evaluating management efforts

Management decisions driven solely by chemical criteria may not benefit biological integrity. Effluent from a wastewater treatment plant on Copper Slough in east-central Illinois, for example, caused significantly lower fish IBI values downstream of the discharge, which had undergone standard secondary treatment with chlorination (Fig. 6b). When chlorine was not present in discharge water, upstream and downstream sites did not differ statistically. Installing expensive tertiary treatment for nitrates, mandated by chemical water quality standards, did not improve the biological status of the stream. In an Ohio study (Yoder and Rankin 1995), fish IBI increased substantially after significant reductions in pollutants, such as oxygen-demanding wastes, suspended solids, ammonia, some nutrients, and in some cases heavy metals and other toxics, in the effluent from two wastewater treatment plants on the Scioto River (Fig. 6c).

Statistical power and precision of IBI

IBI is versatile because it supports use of standard analysis techniques such as ANOVA or *t*-tests designed to evaluate specific hypotheses. Statistical power analysis (Fig. 6d), based on fish data collected by Ohio EPA, demonstrates that IBI can detect six distinct categories of resource condition (Fore et al. 1994). IBI is thus an effective monitoring tool, both to communicate qualitative biological condition and to provide quantitative assessments for use in legal or regulatory contexts.

These examples demonstrate that biological assessment of stream health can be applied in diverse contexts. In contrast, failure to incorporate direct biological evaluations into water resource or fishery management--as in the experiences with tertiary sewage treatment and chlorine-laden effluent--can waste financial resources as well as degrade biological resources.

A BENTHIC IBI FOR THE PACIFIC NORTHWEST

Tests of the IBI concept in recent years demonstrate the appropriateness of an IBI based on benthic invertebrates (B-IBI), including appropriate biological metrics for the Pacific Northwest (Ohio EPA 1988, Karr and Kerans 1991, Kerans and Karr 1994, DeShon 1995, Fore et al. 1996). Macroinvertebrates, such as insects, crustaceans, molluscs, and worms, have many advantages for biomonitoring in the Pacific Northwest. Northwest streams have few fish species (low species richness) but many kinds of macroinvertebrates (Fore et al. 1996). Macroinvertebrate assemblages are ubiquitous, abundant, relatively easy to sample, and encompass taxa with

differential responses to a broad spectrum of human activities. Because the life cycles of some benthic invertebrates extend several years, they are better integrators of past human influences than water chemistry is.

Until recently, few systematic studies have been done on the biota of Pacific Northwest streams, especially with regard to human effects on those systems. In contrast, the impacts of landscape alteration on hydrological processes and water quality are well known (Booth 1991), as are the general impacts of human activity on salmon populations (Bottom 1996) and the effects of urbanization on Lake Washington (Edmundson 1991). The decline of Lake Washington involved an easily defined culprit: nutrient enrichment caused by growing quantities of human waste released into the lake. The solution was simple and specific. King County created a regional sewage management agency to control the problem; the improvement in the lake was rapid and dramatic (Edmundson 1991). But without detailed biological studies of streams, managers and scientists cannot always so clearly convey the full consequences of poorly planned development, or reverse degradation.

The first systematic stream study examined urbanization, the largest threat to streams in the Puget Sound lowlands. A study of 24 Puget Sound streams was launched in 1994 under a grant from the Centennial Clean Water Program of the Washington Department of Ecology; sampling continued in 1995. Benthic invertebrate samples (3 replicates) were collected in each stream from a riffle segment chosen on the basis of cobble size, cover, flow, and slope.

Nine attributes (total, mayfly, stonefly, caddisfly, intolerant, and long-lived taxa richness; percent planaria and amphipod abundance; percent tolerant taxa and percent predator taxa) of stream invertebrates changed systematically along a gradient of human influence, measured as percent impervious area. Aerial and satellite images were analyzed to determine the area within each watershed in each land use. Imperviousness--a measure of a landscape's ability to absorb rainfall--varies from 2% for natural forest, 5% for clearcut areas, 20% for medium-density single-family housing (3-7 units/ha) to 60% for light-industrial and 90% for commercial development (Law 1994). Percent impervious area is directly and linearly related to biological integrity; no threshold of degradation stands out (Fig. 7). Invertebrate assemblages varied more with impervious area (variation among sites, 91%) than among replicate samples from a single site (variation among replicates, 9%). Some biological attributes decline as human influence increases (e.g., total taxa richness and richness of mayflies, stoneflies, caddisflies, and intolerant taxa); other attributes increase (e.g., number of tolerant taxa such as worms).

The best streams in this study (e.g., Rock, Griffin, and Big Anderson Creeks) had B-IBI scores ranging from 35 to 45 (out of 45); the worst streams (Des Moines, Juanita, Kelsey, and Thornton Creeks) had B-IBI scores below 15. B-IBI scores for streams sampled in both years changed very little from one year to the next. Only 2 of 13 sites sampled in both years changed by more than four IBI points; 9 changed by 2 or less.

B-IBI, it seems, can even be related to the health of salmonid populations. For sites with fish data available, a B-IBI above 35 had ratios of juvenile salmon to juvenile cutthroat trout greater than 5; sites scoring below 35 B-IBI had salmon-to-cutthroat ratios of less than 2 (Horner et al. 1986).

In short, macroinvertebrate assemblages are ideal for discerning the impacts of logging (Fore et al. 1996), livestock grazing (Fore et al.,ms), recreation (Patterson 1996), and urbanization (Kleindl 1995, Rossano 1995).

Regardless of locations, four broad classes of metrics--taxa richness and composition, tolerant and intolerant taxa, feeding ecology, and population attributes--reflect gradients of human influence and successfully distinguish least- and most-disturbed sites when benthic invertebrates are used (Fore et al. 1996). Generally, trophic metrics are less reliable for invertebrates than for fishes. Because the characteristics of invertebrate assemblages vary among regions, as do quality of data sets and knowledge of regional faunas, the number of attributes with demonstrated dose-response curves also varies: 13 metrics were used in Tennessee (Kerans and Karr 1994), 10 in logged landscapes of southwest Oregon (Fore et al. 1996), 9 in lowland streams in Puget Sound (Kleindl 1995), 10 in Grand Teton National Park (Patterson 1996), and 11 in a study of 115 streams in west-central Japan (Rossano 1995).

Integrating these studies, I recommend a regionally appropriate IBI with 10 metrics in four groups: (1) taxa richness and composition: total number of taxa; number of mayfly, stonefly, caddisfly, and long-lived taxa; (2) tolerant and intolerant taxa: number of intolerant taxa and number of clinger taxa; proportion of individuals belonging to tolerant taxa; (3) trophic or feeding ecology: proportion of individuals that are predators; and (4) population attributes: dominance (percent) of the three most abundant taxa (Table 3).

In sum, when the multimetric IBI approach is used to assess invertebrate responses along gradients of human influence, the results are strikingly consistent in four geographic areas (Tennessee, Rocky Mountains, Pacific Northwest, and Japan) and for a number of major human impacts (agriculture, grazing, logging, mining, recreation, and urbanization). In addition to conveying information about site condition, IBI, along with other knowledge of sampled watersheds, can speed identification of the causes of degradation and thus suggest management guidelines to prevent further degradation or reverse past damage.

In our Puget Sound study, for example, the B-IBI for one stream (Coal Creek) expected from general watershed condition was higher than the B-IBI we observed. This discrepancy led us to uncover biological effects remaining from an old coal mine in the watershed. The B-IBI from a number of state-agency identified "reference sites" in Wyoming was also lower than expected. A number of these sites, assumed to be minimally influenced by human actions, were then discovered to be biologically degraded. In this case, the existing methods of defining reference sites turned out to be inadequate (Patterson 1996).

As shown by 15 years' experience, when biological indexes, conventional water chemistry, and hydrologic analyses complement one another, water resource protection improves; the results will be more cost effective and better safeguard biological resources (Davis and Simon 1995).

CHANGE AND RISK ASSESSMENT

Risks exist when the health or well-being of human society is threatened by natural events or by the consequences of what humans themselves have done (Mazaika et al. 1995, Commission on Risk Assessment and Risk Management 1996). People may or may not be able to prepare for natural disasters. But if people are wise, they *can* avoid the risks ensuing from their own actions. Unfortunately, widespread evidence of environmental degradation tells us that we have not been wise, that we have not been effectively assessed and protected ourselves from human-induced ecological risks. Rather, we behave as if we were immune to such risks; we assume that ecological risks do not exist or that we can reverse the effects of our actions when faced with a crisis (Karr 1995b).

Ecological risk assessment must be broader than conventional assessment of risks to human health. Its focus must not be exclusively human but, rather, incorporate the integrity of all living systems. Protecting living systems from chemical contamination alone--the focus of conventional risk assessment--is not enough. Integrity encompasses all species, the health of individuals, and the status of regional landscapes. It is these landscapes that supply clean air and water, produce food and fiber, and offer recreation and other amenities that make up the quality of life. Integrity includes both the elements of living systems and the processes that generate and maintain those systems (Karr 1993, 1996). Systems with biological integrity maintain the natural capital assets, both goods and services, needed to sustain human society (Prugh et al. 1995).

In some respects, the term *ecological risk assessment* obscures the real issue. The critical risks we face are *biological*. The laws of physics and chemistry are immutable, but living, biological systems and their supporting elements and processes can be lost. Nonhuman living systems are bellwethers of conditions that threaten humans; they are also vital because human society depends on them. We must be sure that ecological risk assessment deals directly with the threat to human society caused by depletion of Earth's *living* systems.

Multimetric indexes are important because they effectively accomplish the mandate defined by the EPA Science Advisory Board (SAB 1990): "Attach as much importance to reducing ecological risk as is attached to reducing human health risk."

Going beyond the human health consequences of chemical contaminants requires explicit recognition that we can no longer behave as if ecological risks did not exist (Karr 1996). We must improve our understanding and measurement of ecological risks. Multimetric biological assessment, such as that provided by IBI, identifies the condition of living systems, the best primary endpoint to assess environmental quality. Systematic monitoring of biological resource condition can help communities keep the natural assets that support them. It can also be a major step in efforts to restore degraded systems, reversing the trend toward resource damage and depletion that has prevailed during the twentieth century.

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LITERATURE CITED

Allan, J. D., and A. S. Flecker. 1993. Biological diversity conservation in running waters. Bioscience **43**:32-43.

Angermeier, P. L., and J. R. Karr. 1994. Biological integrity versus biological diversity as policy directives. Bioscience **44**:690-697.

Armstrong, J. S. 1967. Derivation of theory by means of factor analysis, or Tom Swift and his electric factor analysis machine. American Statistician **21**:17-21.

Barbour, M. T., J. B. Stribling, and J. R. Karr. 1995. Multimetric approach for establishing biocriteria and measuring biological condition. Pp. 63-77 *in* W. S. Davis and T. P. Simon, editors. Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, FL.

Barbour, M. T., J. Gerritsen, G. E. Griffith, R. Frydenborg, E. McCarron, J. S. White, and M. L. Bastian. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. Journal North American Benthological Society **15**:185-211.

Booth, D. B. 1991. Urbanization and the natural drainage system: impacts, solutions, and prognoses. Northwest Environmental Journal **7**:93-118.

Bottom, D. L. 1996. To till the water: a history of ideas of fisheries conservation. Pp. *in* D. J. Stouder, P. A. Bisson, and R. J. Naiman, editors. Pacific salmon and their ecosystems: status and future options. Chapman and Hall, New York.

Bowen, J. T. 1970. A history of fish culture as related to the development of fishery programs. American Fisheries Society Special Publication **7**:71-93.

Boyle, T. P., G. M. Smillie, J. C. Anderson, and D. R. Beason. 1990. A sensitivity analysis of nine diversity and seven similarity indices. Research Journal Water Pollution Control Federation **62**:749-762.

Chutter, F. M. 1972. An empirical biotic index of the quality of water in South African streams and rivers. Water Research **6**:19-30.

Commission on Risk Assessment and Risk Management. 1996. Risk assessment and risk management in regulatory decision making. Draft report, Commission on Risk Assessment and Risk Management, Washington, DC.

Courtenay, W. R., Jr., and P. B. Moyle. 1992. Crimes against biodiversity: the lasting legacy of fish introductions. *In* Transactions 57th North American Wildlife and Natural Resources Conference. Wildlife Management Institute, Washington, DC.

Davis, W. S. 1995. Biological assessment and criteria: Building on the past. Pp. 15-29 *in* W. S. Davis and T. P. Simon, editors. 1995. Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, Florida.

Davis, W. S., and T. P. Simon, editors. 1995. Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, FL.

DeShon, J. E. 1995. Development and application of the invertebrate community index (ICI). Pp. 217-243 *in* W. S. Davis and T. P. Simon, editors. 1995. Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, FL.

Edmundson, W. T. 1991. The uses of ecology: Lake Washington and beyond. University of Washington Press, Seattle, WA.

Einstein, H. A. 1972. Sedimentation (suspended solids). Pp. 309-318 *in* R. T. Oglesby, C. A. Carlson, and J. McCann, editors. River ecology and man. Academic Press, New York.

Evans, R. 1965. Industrial wastes and water supplies. Journal American Water Works Association **57**: 625-628.

Fausch, K. D., J. R. Karr, and P. R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Transactions American Fisheries Society **113**:39-55.

Fausch, K. D., J. Lyons, P. L. Angermeier, and J. R. Karr. 1990. Fish communities as indicators of environmental degradation. American Fisheries Society Symposium **8**:123-144.

Fore, L. A., J. R. Karr, and L. L. Conquest. 1994. Statistical properties of an index of biological integrity used to evaluate water resources. Canadian Journal Fisheries and Aquatic Sciences **51**:1077-1087.

Fore, L. S., J. R. Karr, and R. W. Wisseman. 1996. Assessing invertebrate responses to human activities: evaluating alternative approaches. Journal North American Benthological Society **15**:212-231.

Fore, L. S., J. R. Karr, and C. Tait. ms. Benthic invertebrate responses to livestock grazing in southeastern Oregon. Submitted to Journal of Range Management.

Gerritsen, J. 1995. Additive biological indices for resource management. Journal North American Benthological Society **14**:451-457.

Gotelli, N. J., and G. R. Graves. 1996. Null models in ecology. Smithsonian Institution Press, Washington, DC.

Green, R. H. 1979. Sampling design and statistical methods for environmental biologists. Wiley and Son, New York, NY.

Hilsenhoff, W. L. 1982. Using a biotic index to evaluate water quality in streams. Technical Bulletin Number 132. Department of Natural Resources, Madison WI.

Horner, R. R., D. B. Booth, A. Azous, and C. W. May. 1996. Watershed determinants of freshwater ecosystem character and functioning: results of 10 years of research in the Puget Sound region. Pp. *in* L. Roesner, editor. Effects of watershed development and management on aquatic ecosystems. Engineering Foundation, New York, NY.

Howarth, R. W. 1991. Comparative responses of aquatic ecosystems to toxic chemical stress. Pp. 169-195 *in* J. Cole, G. Lovett, and S. Findlay, editors. Comparative analyses of ecosystems: patterns, mechanisms, and theories. Springer-Verlag, New York, NY.

Hughes, R. M., and J. R. Gammon. 1987. Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. Transactions American Fisheries Society **116**:196-209.

Hurlbert, S. H. 1971. The nonconcept of species diversity: a critique and alternative parameters. Ecology **52**:577-586.

James, F. C., and C. E. McCullough. 1990. Multivariate analysis in ecology and systematics: panacea or Pandora's box? Annual Review of Ecology and Systematics **21**:129-166.

Johnson, R. W., and R. Paschal. 1995. The limits of prior appropriation. Illahee **11**:40-50.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries **6**(6):21-27.

Karr, J. R. 1990. Biological integrity and the goal of environmental legislation: lessons for conservation biology. Conservation Biology **4**:244-250.

Karr, J. R. 1991. Biological integrity: A long-neglected aspect of water resource management. Ecological Applications **1**:66-84.

Karr, J. R. 1992. Measuring biological integrity: lessons from streams. Pp. 83-104 *in* S. Woodley, J. Kay, and G. Francis, eds. Ecological integrity and the management of ecosystems. St. Lucie Press, Delray Beach, FL.

Karr, J. R. 1993. Defining and assessing ecological integrity. Environmental Toxicology and Chemistry **12**:1521-1531.

Karr, J. R. 1995a. Clean water is not enough. Illahee **11**:51-59.

Karr, J. R. 1995b. Risk assessment: We need more than an ecological veneer. Human and Ecological Risk Assessment **1**:436-442.

Karr, J. R., 1996. Ecological Integrity and ecological health are not the same. Pp. 100-113 *in* P. Schulze, editor. Engineering Within Ecological Constraints. National Academy Press, Washington, DC.

Karr, J. R., and D. R. Dudley. 1981. Ecological perspective on water quality goals. Environmental Management **5**:55-68.

Karr, J. R., and T. E. Martin. 1981. Random numbers and principal components: Further searches for the unicorn. Pp. 20-24 *in* D. Capen, editor. The use of multivariate statistics in studies of wildlife habitat. General Technical Report, Report RM-87. USDA Forest Service,

Karr, J. R., and B. L. Kerans. 1991. Components of biological integrity: their definition and use in development of an integrative IBI. Pp. 1-16 *in* T. P. Simon and W. S. Davis, editors. Environmental indicators: measurement and assessment endpoints. Proceedings of the 1991 Midwest Pollution Control Biologists' Meeting. EPA 905/R-92/003. US Environmental Protectin Agency, Chicago, IL.

Karr, J. R., and L. S. Fore. 1996. Making biological monitoring more effective: integrating biological sampling and analytical procedures to document biological responses to human influence. Final Report to US Environmental Protection Agency. School of Fisheries, University of Washington, Seattle, WA.

Karr, J. R., R. C. Heidinger, and E. H. Helmer. 1985. Sensitivity of the index of biotic integrity to changes in chlorine and ammonia levels from wastewater treatment facilities. Journal Water Pollution Control Federation **57**:912-915.

Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. Assessment of biological integrity in running water: a method and its rationale. Special Publication No. 5. Illinois Natural History Survey, Champaign, IL.

Karr, J. R., P. R. Yant, K. D. Fausch, and I. J. Schlosser. 1987. Spatial and temporal variability of the index of biotic integrity in three midwestern streams. Transactions American Fisheries Society **116**:1-11.

Kerans, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications **4**:768-785.

Kleindl, W. J. 1995. A benthic index of biotic integrity for Puget Sound lowland streams, Washington, USA. Unpublished M. Sc. Thesis, University of Washington, Seattle, WA.

Kolkwitz, R., and M. Marsson. 1908. Okologie der pflanzlichen saprobien. Berichte der Deutschen botanischen Gesellschaft **26a**:505-519. (Translated 1967. Ecology of plant saprobia. Pp. 47-52 *in* L. E. Kemp, W. M. Ingram, and K. M. Mackenthum, editors. Biology of water pollution. Federal Water Pollution Control Administration, Washington, DC.

Law, A. W. 1994. The effects of watershed urbanization on stream ecosystem integrity. M. Sc. Thesis, University of Washington, Seattle, WA.,

Lenat, D. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. Journal North American Benthological Society **7**:222-233.

Lenat, D. 1993. A biotic index for the southeastern United States: derivation and list of tolerance values, with criteria for assigning water quality ratings. Journal North American Benthological Society **12**:279-290.

Ludwig, J. A., and J. F. Reynolds. 1988. Statistical ecology. John Wiley, New York.

Maser, C., and J. R. Sedell. 1994. From the forest to the sea: the ecology of wood in streams, rivers, estuaries, and oceans. St. Lucie Press, Delray Beach, FL.

Mazaika, R., R. T. Lackey, and S. L. Friant, editors. 1995. Ecological risk assessment: Use, abuse, and alternatives. Human and Ecological Risk Assessment **1**:337-458.

Meffe, G. K. 1992. Techno-arrogance and halfway technologies: salmon hatcheries on the Pacific Coast of North America. Conservation Biology **6**: 350-354.

Meyer, J. L., W. H. McDowell, T. L. Bott, J. W. Elwood, C. Ishizaki, J. M. Melack, B. L. Peckarsky, B. J. Peterson, and P. A. Rublee. 1988. Elemental dynamics in streams. J. North American Benthological Society **7**:410-432.

Miller, D. L., P. M. Leonard, R. M. Hughes, J. R. Karr, P. B. Moyle, L. H. Schrader, B. A. Thompson, R. A. Daniels, K. D. Fausch, G. A. Fitzhugh, J. R. Gammon, D. B. Halliwell, P. L. Angermeier, and D. J. Orth. 1988. Regional applications of an index of biotic integrity for use in water resource management. Fisheries **13**(5):12-20.

Mitchell, W. C., and A. F. Burns. 1938. Statistical indicators of cyclical events. National Bureau of Economic Research, New York.

Naiman, R. J., editor. 1992. Watershed management: balancing sustainability and environmental change. Springer-Verlag, New York.

Nehlsen, W., J. E. Williams, and J. A. Lichatowich. 1991. Pacific salmon at the crossroads: stocks of salmon at risk from California, Oregon, Idaho, and Washington. Fisheries **16**(2):4-21.

Norris, R. H. 1995. Biological monitoring: the dilemma of data analysis. Journal North American Benthological Society **14**:440-450.

Ohio EPA (Environmental Protection Agency). 1988. Biological criteria for the protection of aquatic life, volumes 1-3. Ecological Assessment Section, Division of Water Quality Monitoring and Assessment, Ohio EPA, Columbus, Ohio.

Osenberg, C. W., R. J. Schmitt, S. J. Holbrook, K. E. Abu-Saba, and A. R. Flegal. 1994. Detection of environmental impacts: natural variability, effect size, and power analysis. Ecological Applications **4**:16-30.

Patterson, A. J. 1996. The effect of recreation on biotic integrity of small streams in Grand Teton National Park. M. Sc. Thesis, University of Washington, Seattle, WA.

Plafkin, J. L., M. T. Barbour, K. Porter, S. Gross, and R. M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. EPA/440/4-89/001. US Environmental Protection Agency, Washington, DC.

Pringle, C. M., R. J. Naiman, G. Bretschko, J. R. Karr, M. Oswood, J. Webster, R. Welcomme, and M. J. Winterbourn. 1988. Patch dynamics in lotic systems: The stream as a mosaic. Journal North American Benthological Society **7**:503-524.

Prugh, T., R. Costanza, J. H. Cumberland, H. Daly, R. Goodland, and R. B. Norgaard. 1995. Natural capital and human economic survival. ISEE Press, Solomons, MD.

Rapport, D. J., H. A. Regier, and T. C. Hutchinson. 1985. Ecosystem behaviour under stress. American Naturalist **125**:617-640.

Rexstad, E. A., D. D. Miller, C. H. Flather, E. M. Anderson, J. W. Hupp, and D. R. Anderson. 1988. Questionable multivariate statistical inference in wildlife habitat and community studies. Journal Wildlife management **52**:794-798.

Reynoldson, T. B., and D. M. Rosenberg. 1996. Sampling strategies and practical considerations in building reference data bases for prediction of invertebrate community structure. Pp. 1-31 *in* R. C. Bailey, R. H. Norris, and T. B. Reynoldson, editors. Study design and data analysis in benthic macroinvertebrate assessments of freshwater ecosystems using a reference site approach. Ninth Annual Technical Information Workshop, June 4th, 1996. 44th Annual Meeting of the North American Benthological Society, Kalispell, MT.

Rossano, E. M. 1995. Development of an index of biological integrity for Japanese streams (IBI-J). M. Sc. Thesis, University of Washington, Seattle, WA.

Schindler, D. W. 1987. Determining ecosystem responses to anthropogenic stress. Canadian Journal Fisheries and Aquatic Sciences **44** (Suppl. 1):6-25.

Schindler, D. W. 1990. Experimental perturbations of whole lakes as tests of hypotheses concerning ecosystem structure and function. Oikos **57**:25-41.

Schmitt, R. J., and C. W. Osenberg, editors. 1996. Detecting ecological impacts: concepts and applications in coastal habitats. Academic Press, San Diego, CA.

Science Advisory Board (SAB). 1990. Reducing risk: setting priorities and strategies for environmental protection. SAB-EC-90-021. US Environmental Protection

Agency, Washington, DC.

Simon, T. P., and J. Lyons. 1995. Application of the index of biotic integrity to evaluate water resource integrity in freshwater ecosystems. Pp. 245-262 *in* W. S. Davis and T. P. Simon, editors. 1995. Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, Florida.

Sioli, H. 1975. Tropical rivers as expressions of their terrestrial environments. Pp. 275-288 in F. B. Golley and E. Medina, eds. Tropical ecological systems: trends in terrestrial and aquatic research. Springer-Verlag, New York.

Stewart-Oaten, A. 1995. Rules and judgements in statistics: three examples. Ecology **76**:2001-2009.

Stewart-Oaten, A., W. W. Murdoch, and K. R. Parker. 1986. Environmental impact assessment: "pseudoreplication in time? Ecology **67**:929-940.

Stewart-Oaten, A. 1996. Goals in environmental monitoring. Pp. 17-28 *in* R. J. Schmitt and C. W. Osenberg, editors. 1996. Detecting ecological impacts: concepts and applications in coastal habitats. Academic Press, San Diego, California.

Suter, G. W., II. 1993. A critique of ecosystem health concepts and indexes. Environmental Toxicology and Chemistry **12**:1533-1539.

Underwood, A. J. 1991. Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. Australian Journal of Marine and Freshwater Research **42**:569-587.

Underwood, A. J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. Ecological Applications **4**:3-15.

USEPA (US Environmental Protection Agency). 1990. Biological criteria: national program guidance for surface waters. EPA 440-5-90-004. USEPA, Office of Water Regulations and Standards, Washington, DC.

USEPA (US Environmental Protection Agency). 1996a. Summary of state biological assessment programs for streams and rivers. EPA 230-R-96-007. USEPA, Office of Policy, Planning, and Evaluation, Washington, DC.

USEPA (US Environmental Protection Agency). 1996b. Biological assessment methods, biocriteria, and biological indicators: bibliography of selected technical, policy, and regulatory literature. EPA 230-B-96-001. USEPA, Office of Policy, Planning, and

Evaluation, Washington, DC.

Weber, C. I. 1981. Evaluation of the effects of effluents on aquatic life in receiving waters--an overview. Pp. 3-13 *in* J. M. Bates and C. I. Weber, editors. Special Publication No. 730. ASTM (American Society of Testing and Materials), Philadelphia, PA.

White, R. J., J. R. Karr, and W. Nehlsen. 1995. Better roles for fish stocking in aquatic resource management. American Fisheries Society Symposium **15**:527-547.

Wilhm, J. L., and T. C. Dorris. 1968. Biological parameters for water quality criteria. Bioscience **18**:477-481.

Wolda, H. 1981. Similarity indices, sample size, and diversity. Oecologia **50**:296-302.

Yoder, C. O., and E. T. Rankin. 1995. Biological response signatures and the area of degradation value: new tools for interpreting multimetric data. Pp. 263-286 *in* W. S. Davis and T. P. Simon, editors. 1995. Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers, Boca Raton, Florida.

Table 1. Five primary classes of water resource attributes altered by the cumulative effects of human activity, with examples of degradation in Northwest watersheds (from Karr 1995a).

Attribute	Components	Degradation in Northwest watersheds
Water quality	Temperature; turbidity, dissolved oxygen; acidity; alkalinity; heavy metals, toxic substances; organic and inorganic chemicals.	Increased temperature Oxygen depletion Chemical contaminants
Habitat structure	Substrate type; water depth and current speed; spatial and temporal complexity of physical habitat	Sedimentation and loss of spawning gravel Lack of coarse woody debris Destruction of riparian vegetation and banks Lack of deep pools Altered distribution of constrained and unconstrained channel reaches
Flow regime	Water volume; timing of flows	Altered flows limiting survival of salmon and other aquatic organisms at various phases in their life cycles
Food (energy) source	Type, amount, and size of organic particles entering stream; seasonal pattern of energy availability	Altered supply of organic material from riparian corridor Reduced or unavailable nutrients from the carcasses of adult salmon after spawning
Biotic interactions	Competition; predation; disease; parasitism; mutualism	Increased predation on young by native and exotic species Overharvest by sport and commercial fishers

Table 2. Metrics used to assess fish communties in the midwestern United States (from Karr 1981 and Fausch et al. 1984) and the Willamette River, Oregon (from Hughes and Gammon 1987).

Category	Midwestern IBI metric	Oregon IBI metric
Species composition and abundance	1. Total number of native fish species	Same
	2. Number of darter species	Number of sculpin species
	3. Number of sunfish species	Number of minnow species
	4. Number of sucker species	Same
	5. Number of intolerant species	Same
	6. Proportion of green sunfish individuals	Proportion of common carp individuals
Trophic composition	7. Proportion of omnivore individuals	Same
	8. Proportion of insectivorous cyprinid individuals	Proportion of insectivore individuals
	9. Proportion of top carnivore individuals	Proportion of catchable salmonid individuals (> 20 cm)
Fish abundance and condition	10. Number of individuals in sample	Same
	11. Proportion of hybrid individuals	Proportion of individuals from introduced species
	12. Proportion of individuals with disease, tumors, fin damage, or skeletal anomalies	Same
	13. None	Total fish biomass

Table 3. Metrics selected for inclusion in a Pacific Northwest benthic invertebrate index of biological integrity (B-IBI) based on six studies of invertebrate responses to human activities. An X in the column indicates that the metric varied systematically along a gradient of human impact for that data set; blanks indicate that the metric did not vary systematically; -- indicates that the metric was not tested for that data set. Sources: Tennessee (TENN), Kerans and Karr 1994; southwestern Oregon (SWOR), Fore et al. 1996; southeastern Oregon (SEOR), Fore et al. ms.; Puget Sound (PUSD), Kleindl 1995; Japan (JAPN), Rossano 1995; Grand Teton National Park (GTNP), Patterson 1996.

		Geographic Area					
Metric	Predicted response	TENN	SWOR	SEOR	PUSD	JAPN	GTNP
Taxa richness and composition							
Total number of taxa	Decrease	X	X	X	X	X	X
Number of Ephemeroptera taxa	Decrease	X	X		X	X	X
Number of Plecoptera taxa	Decrease	X	X	X	X		X
Number of Trichoptera taxa	Decrease	X	X	X	X	X	X
Number of long-lived taxa	Decrease	--	X	X	X	--	
Tolerance							
Number of intolerant taxa	Decrease	X	X	X	X	X	X
% of individuals in tolerant taxa	Increase	X	X		X	X	X
Feeding ecology							
% of predator individuals	Decrease	X		X	X		X
Number of clinger taxa	Decrease	--	--	--	--	X	--
Population attributes							
% dominance (2 or 3 taxa)	Increase	X	X				X

FIGURE CAPTIONS

Figure 1. Generalized dose-response curves for toxicology (upper panel) and cumulative ecological effects of human activity (middle and lower panels). See text for explanation.

Figure 2. Ecological "dose-response" curves across a gradient of human influence for four attributes of invertebrate assemblages. The human influence gradient integrates type and amount of effluent; presence of dams, weirs, and levees; and the condition of the riparian corridor for 65 streams in west-central Japan (from Rossano 1995).

Figure 3. Lines of maximum species richness as a function of stream order, based on historical data from midwestern streams (from Fausch et al. 1984). These lines show that species richness varies predictably across regions and stream orders, generally in two groups: forested regions in the eastern Midwest (upper 6 lines) and Great Plains (lower 2 lines).

Figure 4. Median scores for benthic IBI in riffles of the North Fork Holston River from 1973 to 1976. Arrow indicates location of a streamside sludge pond (from Kerans and Karr 1994).

Figure 5. **A.** Fish IBI scores for ten stations during 1978-80 on Big Ditch, a channelized, third-order stream in east-central Illinois (see text for discussion; from Karr et al. 1986). Shaded bars indicate groups of sites for which means are statistically indistinguishable ($p<0.05$, Student-Newman-Keuls test). Vertical line through each station bar shows the range; the bar itself extends one standard deviation above and below the mean (horizontal line). **B.** Integrity classes as revealed by IBI for each township in the Raisin River Watershed, Michigan (from Fausch et al. 1984). Areas with low IBI values are associated with larger towns, extensive agricultural areas, and feedlots. IBI values range as follows: excellent (E), 56-60; good (G), 48-52; fair (F), 40-44; poor (P), 28-34; very poor (VP), 12-22; no fish (NF), no IBI value. **C.** Distribution of sites by IBI classes in six midwestern regions or watersheds (from Karr et al. 1986). Integrity classes as defined in B.

Figure 6. **A**. Changes in IBI total scores over time in Wertz Drain at Wertz Woods, Allen County, Indiana (from Karr et al. 1986). Sedimentation in Wertz Woods caused both habitat quality and the resident fish community to deteriorate. IBI scores clearly indicate this decline followed by a slow improvement. **B.** IBI total scores for stations upstream and downstream of wastewater treatment effluent in Copper Slough, east-central Illinois. Phase I, standard secondary treatment; phase II, secondary treatment without chlorination; phase III, secondary treatment without chlorination but with tertiary denitrification. **C.** Longitudinal trend in IBI for the Scioto River, Ohio, downstream from Columbus, 1979 and 1991. Arrowheads indicate locations of wastewater treatment plants (WWTP) and combined sewer overflow (CSO). Dashed lines indicate standards for designated uses: WWH, warmwater habitat; EWH, excellent warmwater habitat. **D.** Power curves for IBI estimated from nine locations sampled three times. Actual points are shown only for $\alpha = 0.05$; other values of α are shown as smoothed lines.

Figure 7. Relationship between B-IBI and percent impervious area for streams sampled in the Puget Sound lowlands.